

Application of LCIA and comparison of different EOL scenarios: The case of used lead-acid batteries

Stavros E. Daniel, Costas P. Pappis*

University of Piraeus, Department of Industrial Management, 80 Karaoli & Dimitriou Street, 185 34 Piraeus, Hellas, Greece

Received 17 August 2006; received in revised form 28 November 2007; accepted 4 December 2007

Available online 3 March 2008

Abstract

Recent years have seen the development of significant research effort focusing on methodological issues regarding the analysis of environmental policies towards integrated pollution intervention and control. Such policies concerning the reverse supply chain of used products have different implications for the environment, which need to be assessed.

In this paper, the environmental design of industrial products (EDIP) method, adapted to the Society of Environmental Toxicology and Chemistry's life cycle impact assessment (SETAC LCA) method, is used in order to assess the environmental implications in the case of two alternative EOL scenarios, namely, disposal and recovery of used lead-acid batteries. From the comparison of the two scenarios, conclusions are drawn regarding the environmental profile of the examined channels and the findings are analysed and discussed.

© 2007 Elsevier B.V. All rights reserved.

Keywords: Life cycle impact assessment; End-of-Life (EOL) scenarios; EDIP method; Reverse supply chain; Lead-acid battery

1. Introduction

An important phase in a typical life cycle analysis (LCA) is impact assessment (IA), which follows the inventory of the inputs and outputs of the examined product systems. In this phase, characterization and assessment of effects associated with a product, process or activity takes place. In particular, the methodology of life cycle IA comprises four steps, namely, classification, characterization or calculation, normalization and valuation or weighting (Groupe des Sages, 1997; Hertwich and James, 2001; ISO 14042:2000, 2000; Lindfors et al., 1995a,b; Miettinen and Hamalainen, 1997; Wenzel et al., 1997). Life cycle IA may be quantitative and/or qualitative and should address effects such as environmental and health considerations, habitat modification and other impacts (NORD, 1995; Schmidt-Bleek, 1994; SETAC, 1991; UNECE, 1991a,b).

Special interest has been focused recently on Life cycle IA. This phase is a technical procedure, which characterizes and evaluates the impacts of environmental burdens identified in the

previous phase of LCA, namely, inventory analysis. During this phase, it is also attempted to identify related hazards, thus assisting manufacturers to prioritize areas for action in order to get the best results for their investments (Berkhout, 1995; Curran, 1993; Lee et al., 1995). For the evaluation of environmental effects several methods have been proposed (Hanssen, 1998; Hertwich et al., 1997; Krozer and Vis, 1998). Some of them produce one index representing the total environmental effect of the examined product system while other methods express the environmental effects with more than one index. For example, the methods of critical volume or MIPS, which give their results in one endpoint (ISO 14040:1997, 1997; ISO 14041:1998, 1998). Included in the methods proposed are the health hazard scoring (HHS) system (Srinivasan et al., 1992), the material input per service-unit (MIPS) (Bringezu et al., 1994; Hinterberger, 1994; SETAC, 1993), the Swiss ecopoint (SEP) method (Abhe et al., 1990), the sustainable process index (SPI) (Narodoslawsky and Krotscheck, 1995; Sage, 1993), the environmental priority strategies (EPS) (Hanssen et al., 1994; Rydh, 1999; UNECE, 1994), the Society of Environmental Toxicology and Chemistry's life cycle impact assessment (SETAC LCA) (ISO 14042:2000, 2000), the method of critical volume (Heijungs et al., 1992), the eco-indicator method (Goedkoop, 1995) and the Tellus method (Tellus Institute, 1992; Tørsløv, 2005).

* Corresponding author. Tel.: +30 210 9422052; fax: +30 210 4142328.

E-mail addresses: stdanielgr@yahoo.com (S.E. Daniel), pappis@unipi.gr (C.P. Pappis).

A method to get a more accurate interpretation of parameters, which influence the eco-balance, and probably has gained the widest acceptance, is the use of so-called “impact categories and equivalency factors” which is based on SETAC LCA method (Hertwich et al., 1997). In this method, different approaches as the above are used according to the type of the environmental problem. Environmental impacts may be classified in three major impact categories (resources consumption, ecological impacts and impacts on the working environment) and may be global, regional or local.

Our purpose in this paper is to present how an LCIA method, such as the EDIP method, can be applied in assessing alternative EOL scenarios (in our case, two alternative EOL scenarios of lead-acid batteries), to discuss the issues raised and draw conclusions from such an application. This may assist environmental engineers to draw some useful conclusions in relation to the assessment of environmental profiles of EOL scenarios and, more generally, supply chain scenarios.

More specifically, in the present study, the “distance to target” principle is mainly applied for valuation, based on the political targets, as defined by the relevant regulations for each environmental impact category, which hold globally, in EU and Greece. It must be noted that, the exchanges with the environment from the various processes in the systems examined were assessed and the most significant resources consumption, ecological impacts and impacts on the working environment were designated on the basis of the weighting criteria applied in the EDIP method (Hauschild et al., 1998; Wenzel et al., 1997). The EDIP method represents an operationally determined method, which covers all the phases as these are defined by SETAC method (SETAC series of reports) as well as it is harmonized with ISO standards 14042. Additionally, in this paper updated normalization and weighting factors are used based on an extension of the original EDIP methodology in the form of normalization references and weighting factors for EU-15 and the world (Busch, 2005; Stranddorf et al., 2005; Tørsløv, 2005).

In the following chapters IA is applied to the alternative EOL scenarios of lead-acid batteries, i.e. the reverse supply (recycling) or disposal chain of used batteries. The analysis focuses on the categories, which are significant in terms of potential environmental impacts on resources, ecosystems and working environment. Results are presented analytically in one of the impact categories in order to show how the

method is applied. Analytical results and technical details for each impact category are presented in Daniel and Pappis (2003a,b).

2. Ecological impacts

2.1. Global warming

Greenhouse effect as a result of global warming belongs to the main environmental impact categories of “ecological impacts”. It is evident that these impacts influence the eco-balance globally. Applying the above methodology of IA, the classification of the substances that potentially contribute to global warming, is, first of all, taking place. Potential contributors to global warming are substances which:

- (i) absorb infrared radiation and have an atmospheric lifetime sufficient to allow a significant contribution to warming, or
- (ii) have a petrochemical origin and are degraded with release of carbon dioxide (CO₂).

For CO₂, only emissions which result in a net contribution to the atmospheric content of the substance should be included (CO₂ emissions which do not amount to net contributions are the emissions of biofuels such as straw, wood or biogas, which merely utilize some of the existing flows of the earth’s carbon cycle) (Andersson-Skold et al., 1992; Houghton et al., 1992, 1990).

For the characterization of the potential contribution from a given substance, a characterization factor is used defined as the global warming potential (GWP), for a time horizon of 100 years. The emissions, which are characterized, are expressed as equivalent emissions of the reference substance CO₂.

The total global warming potential of the global emissions of greenhouse gases in a specific year (1994) is used as a reference for normalization of the emissions. In Table 1, the actual global emissions of greenhouse gases which are used as normalization references for global warming are summarized and their potential environmental impacts are calculated for the reference year 1994. The normalization reference is given as the average global warming potential per person in the world. For a total global emission of greenhouse gases (1994 figures) equivalent to 4.59×10^{10} kt CO₂/year and a total global population (1994) of 5.29 billion people (UNECE, 1991a,b), the person-

Table 1
Global emissions of some important greenhouse gases and equivalent CO₂ emissions (Ayres, 1995)

Substance	GWP _{100year} (kg CO ₂ /kg substance)	Actual global emission 1990 (kt)	Equivalent CO ₂ emission (kt CO ₂ -equiv./year)
Carbon dioxide	1	2.71×10^7	2.71×10^7
Methane	25	3.51×10^5	8.78×10^6
Nitrous oxide	320	7.23×10^3	2.31×10^6
CFC11	4000	298	1.19×10^6
CFC12	8500	363	3.09×10^6
CFC113	5000	147	7.35×10^5
Carbon monoxide	2	9.96×10^5	1.99×10^6
Total			4.59×10^{10}

Table 2

Actual emissions (1997) of the observed greenhouse gases and equivalent CO₂ emissions of the batteries reverse supply and disposal chain

Substance	GWP _{100year} (kg CO ₂ /kg substance)	Actual emission (kg/year)		Equivalent CO ₂ emission (kg CO ₂ -equiv./year)	
		Reverse supply chain	Disposal chain	Reverse supply chain	Disposal chain
Carbon dioxide	1	7.85×10^8	0	7.85×10^8	0
Carbon monoxide	2	3.21×10^4	9.67×10^4	6.42×10^4	19.3×10^4
Nitrogen oxides	4	3.88×10^4	4.48×10^4	15.5×10^4	17.9×10^4
VOC	2	1.41×10^4	1.34×10^4	2.82×10^4	2.68×10^4
Formaldehyde	2	74.7	0	149.4	0
HC	3	112	0	336	0
Total				7.85×10^8	3.98×10^5

equivalent for global warming based on actual emissions is 8.7 t CO₂-equiv./person/year.

In Table 2 the results of the characterization step of IA for both the batteries reverse supply and disposal chain concerning global warming are presented. The total actual emissions of the batteries reverse chain correspond to 0.148 kg CO₂-equiv./year person-equivalent emissions. So, the normalization factor is calculated by the following equation and expresses the fraction of batteries reverse supply chain contribution in global warming

$$NF_{GW} = \frac{E_{GW}}{E_{GWW}} = \frac{0.148}{8.7 \times 10^3} = 1.7 \times 10^{-5} \text{ equiv./person} \quad (1)$$

where NF_{GW} represents the normalization factor in person equivalents, E_{GW} represents the person-equivalents to global warming based on the actual emissions of the batteries' reverse chain in a year (1997), and E_{GWW} represents the person-equivalents to global warming based on actual emissions in the reference year (1994) for the whole world.

In a similar way, the normalization factor for the batteries disposal chain corresponds to 8.65×10^{-9} equiv./person.

A weighting factor is then derived for use in the quantitative weighting of the potential contribution to the global warming of the examined EOL scenarios against potential contributions to other environmental impact categories. This factor is defined as the ratio of the global warming potential for the actual emissions produced during the reference year (1994) to the global warming potential for the target emissions in the year 2004.

If the expected reduction of the global emissions of greenhouse gases is estimated at 10.7% according to the Organization of the United Nations (Global Environment Facility), this corresponds to a reduction of the emissions per person from 8.7 to 7.77 t CO₂-equiv./year

$$WEP_{GW} = \frac{E_{GW94}}{E_{GWT2004}} = \frac{8.7}{7.77} = 1.12 \quad (2)$$

where WEP_{GW} represents the weighting factor or weighted environmental impact potential for global warming, E_{GW94} represents the person-equivalents to global warming based on the global actual emissions of greenhouse gases in the reference year (1994), and $E_{GWT2004}$ represents the person-equivalents to global warming based on the expected global target emissions of greenhouse gases in the target year (2004).

The final weighting can take place by multiplying the normalized impact potential NF_{GW} of the batteries reverse supply chain with the weighting factor WEP_{GW} . The weighted impact potential WP_{GW} is therefore calculated as follows

$$\begin{aligned} WP_{GW} &= NF_{GW} * WEP_{GW} = 1.7 \times 10^{-5} * 1.12 \\ &= 1.90 \times 10^{-5} \text{ equiv./person} \end{aligned} \quad (3)$$

Correspondingly to Eq. (3), the weighted impact potentials for the batteries disposal chain correspond to 9.69×10^{-9} equiv./person.

2.2. Stratospheric ozone depletion

The potential contribution to stratospheric ozone depletion caused by emissions resulting from the processes taking place in the examined EOL scenarios is assessed following the above IA methodology for global warming.

In the classification step, substances with potentials for contributing to depletion of stratospheric ozone are those which:

- are stable in the atmosphere such they can reach the stratosphere to a significant extent, and
- contain chlorine or bromine which, on release into the stratosphere, assist in a chemical breakdown of ozone.

Significant substances that contribute naturally in the breakdown of ozone are methane (CH₄), nitrous oxide (N₂O), water vapor (H₂O), and chlorine and bromine compounds. The breakdown occurs via chemical reactions. There are reactions, which regenerate all of the most significant ozone depleting substances, so that one molecule can participate in the breakdown of many ozone molecules. Human activities have caused an increase in the quantity of substances (chlorine and bromine compounds, CFCs, etc.) in the stratosphere, which contribute to chemical breakdown of ozone, especially over the past 40 years. A common feature of all these substances is that they are chemically stable in the lower parts of the atmosphere and they survive long enough to reach the stratosphere (Solomon and Albritton, 1992).

The equivalency factors are calculated with the help of a characterization factor defined as the ozone depletion potential (ODP). This factor expresses the expected contribution of a substance to stratospheric ozone depletion in relation to ozone depletion on emission of an equivalent quantity of the reference

substance, i.e. CFC11. This substance was chosen because it has been well studied and is one of the most important ozone depleting substances.

The actual global emissions for substances contributing to stratospheric ozone depletion for the reference year (1994) are the global man-made emissions of halocarbons as shown in Table 4, which can be used as a reference for normalization of the emissions. The normalization reference is given as the average stratospheric ozone depletion per person in the world. For a total global emission of halocarbons (1994 figures) equivalent to 544.87 kt CFC11/year and a total global population (1994) of 5.29 billion people, the person-equivalent for stratospheric ozone depletion based on actual emissions is 103 g CFC11-equiv./person/year.

The inventoried results for both the EOL scenarios of the batteries reverse supply and disposal chain show that there were not any substances contributing to stratospheric ozone depletion. Thus, it is not necessary to continue the evaluation of the potential environmental impacts of the examined chains. However, it is purposeful to evaluate the importance of this environmental category by calculating its weighting factor. This can be derived for use in the quantitative weighting of the potential contribution to stratospheric ozone depletion against potential contributions to other environmental impact categories. This factor is defined as the ratio of the ozone depletion potential for the actual emissions produced during the reference year (1994) to the ozone depletion potential for the target emissions in the year 2004.

If the expected reduction of the global emissions of substances contributing to stratospheric ozone depletion is estimated at 98.41% according to global reduction targets for the industrialized countries, this corresponds to a reduction of the emissions per person from 103 g to 1.63 g CFC11-equiv./year

$$WEP_{ODP} = \frac{E_{ODP94}}{E_{ODPT2004}} = \frac{103}{1.63} = 63 \quad (4)$$

where WEP_{ODP} represents the weighting factor or weighted environmental impact potential for stratospheric ozone depletion, E_{ODP94} represents the person-equivalent to stratospheric ozone depletion based on the global actual emissions of substances contributing to ozone depletion in the reference year (1994), and $E_{ODPT2004}$ represents the person-equivalent to stratospheric ozone depletion based on the expected global target emissions of substances contributing to ozone depletion in the target year (2004).

2.3. Photochemical ozone formation

Assessment of the contribution of the examined EOL scenarios to the increase of air concentrations of reactive substances, photo-oxidants, which are injurious to the health of living organisms in the lower stratum of atmosphere (troposphere) is the objective of this impact category.

As characterization factor the potential of photochemical ozone creation (POCP) in a period of 4–9 days is defined. The ozone formation potential is defined only for VOCs. Given the significance of the presence of the NO_x for ozone formation, two sets of POCP for “low NO_x ” areas and “high

NO_x ” areas are created. The impact potentials are expressed as an equivalent emission of the reference substance ethylene (C_2H_4).

As normalization reference the total impact potentials of the actual European emissions in the reference year (1994) are used. This reference is presented as the average ozone formation potential per inhabitant in the area inventoried.

The European person-equivalent for photochemical ozone formation in 1994 corresponds to 25 kg C_2H_4 -equiv./person/year, which is based on the extension of the original EDIP methodology. This results to 2.96×10^{-5} kg C_2H_4 -equiv./year regarding the batteries reverse chain. So, the normalization factor results by the following equation and expresses the fraction of the contribution in photochemical ozone formation by the batteries reverse chain

$$\begin{aligned} NF_{POCP} &= \frac{E_{POCP}}{E_{POCPEU}} = \frac{2.96 \times 10^{-5}}{25} \\ &= 1.18 \times 10^{-6} \text{ equiv./person} \end{aligned} \quad (5)$$

where NF_{POCP} represents the normalization factor in person equivalents, E_{POCP} represents the person-equivalents to photochemical ozone formation based on the actual emissions of the batteries reverse chain in a year (1997), and E_{POCPEU} represents the person-equivalents to photochemical ozone formation based on actual emissions in the reference year (1994) for the European Union.

According to Eq. (5), the normalization factor for batteries disposal chain corresponds to 1.36×10^{-6} equiv./person.

A weighting factor is then derived for use in the quantitative weighting of the potential contribution to the photochemical ozone formation against potential contributions to other environmental impact categories. This factor is defined as the ratio of the POCP for the actual emissions produced during the reverse chain of the batteries to the photochemical ozone formation potential for the target emissions in the year 2004.

If the expected reduction of the emissions of substances contributing to photochemical ozone formation is estimated at 98.41%, according to reduction targets defined by UNECE's protocols, this corresponds to a reduction of the emissions per person from 25 kg C_2H_4 -equiv./person/year to 18.8 kg C_2H_4 -equiv./person/year.

A weighting factor is then derived for use in the quantitative weighting of the potential contribution to ozone formation against potential contributions to other environmental impact categories. This factor is defined as the ratio of the EU potential for ozone formation during the reference year (1994) to the ozone formation potential for the target emissions in the year 2004

$$WEP_{POCP} = \frac{E_{POCP94}}{E_{POCPT2004}} = \frac{25}{18.8} = 1.33 \quad (6)$$

where WEP_{POCP} represents the weighting factor or weighted environmental impact potential for ozone formation, E_{POCP94} represents the person-equivalents to ozone formation based on the European actual emissions in the reference year (1994), and $E_{POCPT2004}$ represents the person-equivalents to ozone forma-

tion based on the European target emissions in the target year (2004).

The final weighting can now be made by multiplying the normalized impact potential NF_{POCP} of the batteries' reverse chain by the weighting factor WEP_{POCP} and the weighted impact potential WP_{POCP} is therefore calculated as

$$\begin{aligned} WP_{POCP} &= NF_{POCP} * WEP_{POCP} = 1.18 \times 10^{-6} * 1.33 \\ &= 1.57 \times 10^{-6} \text{ equiv./person} \end{aligned} \quad (7)$$

Correspondingly to Eq. (7), the weighted impact potential for the disposal chain is equal to 1.81×10^{-6} equiv./person.

2.4. Acidification

In this impact category, the assessment focuses in substances, which contribute to effects of acidification of the environment observed in the form of a pronounced lack of health especially among conifers in many forests, extinction of fishes in lakes or the loss of irreplaceable historic monuments in the industrialized world.

As equivalency factor for use in the calculation of impact potentials the reference substance sulfur dioxide (SO_2) is used, which is the most important man-made acidifying substance. From the inventory analysis of the batteries reverse supply and disposal chain, the following impact assessment table is obtained.

Given a total emission of EU countries of 23800 kt SO_2 -equivalents as this results from the calculation of the most important acidification substances in the reference year of 1994, the normalization reference for acidification based on the EU countries is 74 kg SO_2 -equiv./person/year.

The total actual emissions (Daniel and Pappis, 2003a,b) of the batteries reverse supply chain correspond to 2.18×10^{-4} kg SO_2 -equiv./year person equivalent emissions. The normalization factor is obtained from the following equation and expresses the fraction of the contribution in acidification by the batteries reverse supply chain

$$NF_{AC} = \frac{E_{AC}}{E_{ACEU}} = \frac{2.18 \times 10^{-4}}{74} = 2.94 \times 10^{-6} \text{ equiv./person} \quad (8)$$

where NF_{AC} represents the normalization factor in person equivalents, E_{AC} represents the person-equivalent to acidification based on the actual emissions of the batteries reverse supply chain in a year (1997), and E_{ACEU} represents the person-equivalent to acidification based on actual emissions in the reference year (1994) for EU.

According to Eq. (8), the normalization factor for the batteries disposal chain corresponds to 1.13×10^{-4} equiv./person.

As the contribution of Greece to acidification is not available, the weighting factor for this impact category is based on the EU reduction target for the reference year 2004. A reduction target emission of 18740.16 kt SO_2 -equiv./year results based on a 21.26% reduction of the total emissions, which corresponds to 58.27 kg SO_2 -equiv./person/year.

The weighting factor is thus obtained from the following equation

$$WEP_{ANC} = \frac{E_{ANC94}}{E_{ANCT2004}} = \frac{74}{58.27} = 1.27 \quad (9)$$

where WEP_{ANC} represents the weighting factor or weighted environmental impact potential for acidification, E_{ANC94} represents the person-equivalents to acidification based on the EU actual emissions in the reference year (1994), and $E_{ANCT2004}$ represents the person-equivalents to acidification based on the EU target emissions in the target year (2004).

The final weighting can now be made by multiplying the normalized impact potential NF_{ANC} of the batteries' reverse chain by the weighting factor WEP_{ANC} . The weighted impact potential WP_{ANC} is therefore calculated as

$$\begin{aligned} WP_{ANC} &= NF_{ANC} * WEP_{ANC} = 2.94 \times 10^{-4} * 1.27 \\ &= 3.73 \times 10^{-6} \text{ equiv./person} \end{aligned} \quad (10)$$

Correspondingly to Eq. (10), the weighted impact potential for the disposal chain is equal to 1.44×10^{-4} equiv./person.

2.5. Nutrient enrichment

Nutrient enrichment is defined as the man-made impact on aquatic or terrestrial systems of nitrogen (N), or phosphorous (P). Substances with the potential to contribute to nutrient enrichment are therefore substances containing nitrogen and/or phosphorous. The most significant sources of nutrient enrichment are the agricultural use of fertilizers (NO_3^- and NH_3), the emission of oxides of nitrogen (NO_x) and the contents of N and especially P in wastewater from households and industry. In the inventory analysis of the batteries reverse supply and disposal chain are noted only emissions of oxides of nitrogen.

In the characterization step, three equivalency factors can be used. These concern:

- (i) a N potential, which expresses the substance's nitrogen content,
- (ii) a P potential, which expresses the substance's phosphorous content, and
- (iii) an equivalency factor for a total nutrient enrichment potential, where the nitrogen and phosphorous contents are aggregated in a figure based on the assumption of an average ratio of 16:1 between the nitrogen and phosphorous contents in aquatic organisms.

In this study the total nutrient enrichment potential (NE) is adopted. Given a total emission of EU countries of 21467 kt NO_3^- -equivalents as this results from the calculation of the most important substances contributing to nutrient enrichment in the reference year of 1994, the normalization reference for nutrient enrichment is 58 kg NO_3^- -equiv./person/year.

The above total actual emissions of the batteries reverse supply chain corresponds to 1.42×10^{-4} kg NO_3^- -equiv./year person equivalent emissions (Daniel and Pappis, 2003a,b). So, the normalization factor is obtained from the following equa-

tion and expresses the fraction of the contribution in nutrient enrichment by the batteries reverse supply chain

$$\begin{aligned} \text{NF}_{\text{NE}} &= \frac{E_{\text{NE}}}{E_{\text{NEEU}}} = \frac{1.42 \times 10^{-4}}{58} \\ &= 2.44 \times 10^{-6} \text{ equiv./person} \end{aligned} \quad (11)$$

where NF_{NE} represents the normalization factor in person equivalents, E_{NE} represents the person-equivalent to nutrient enrichment based on the actual missions of the batteries reverse supply chain in a year (1997), and E_{NEEU} represents the person-equivalent to nutrient enrichment based on actual emissions in the reference year (1994) for EU.

According to Eq. (11), the normalization factor for batteries disposal chain corresponds to 2.82×10^{-6} equiv./person.

Using political reduction targets (UNECE, 1991a,b) of the substances containing nitrogen and phosphorous of a 20% for the target year 2004 the following weighting factor is obtained

$$\text{WEP}_{\text{NE}} = \frac{E_{\text{NEEU94}}}{E_{\text{NET2004}}} = \frac{58}{47.54} = 1.22 \quad (12)$$

where WEP_{NE} represents the weighting factor or weighted environmental impact potential for nutrient enrichment, E_{NEEU94} represents the person-equivalents to acidification based on the EU countries actual emissions in the reference year (1994), and E_{NET2000} represents the person-equivalents to nutrient enrichment based on the EU countries target emissions in the target year (2004).

The final weighting can now be obtained by multiplying the normalized impact potential NF_{NE} of the batteries reverse supply chain by the weighting factor WEP_{NE} . The weighted impact potential WP_{NE} is therefore calculated as

$$\begin{aligned} \text{WP}_{\text{NE}} &= \text{NF}_{\text{NE}} * \text{WEP}_{\text{NE}} = 2.44 \times 10^{-5} * 1.22 \\ &= 2.98 \times 10^{-6} \text{ equiv./person} \end{aligned} \quad (13)$$

Correspondingly to Eq. (13), the weighted impact potential for the disposal chain is equal to 3.44×10^{-6} equiv./person.

2.6. Ecotoxicity–human toxicity

The main difference in this impact category from other such categories is that there are no established ecotoxicity or human toxicity factors to be adopted for use as equivalency factors. This category is characterized by a large collection of different mechanisms such as inhibition of specific enzymes in certain organisms, alteration of heritable characteristics (genotoxicity), or a more general narcotic effect in the case of ecotoxicity. On the other hand, chemicals emitted as a consequence of human activities can contribute to human toxicity via exposure through the environment.

The impact potentials for ecotoxicity or human toxicity are normalized in the following three general toxic impact categories mainly according to the geographic scale, which the impacts can affect, and the time scale, over which are active. These three aggregated toxicity impact categories concern:

- Persistent toxicity (ecotoxicity or human toxicity on a regional scale). This includes chronic ecotoxicity in water (etwc) and soil (etsc) and human toxicity in water (htw) and soil (hts).
- Ecotoxicity on a local scale. This includes acute ecotoxicity in water (etwa) and ecotoxicity to the microorganisms in waste water treatment plants (etp).
- Human toxicity in air (hta), on a local scale.

Normalization is taking place by dividing the product's impact potentials by the corresponding normalization references. Then, aggregation of the normalized potentials for ecotoxicity and human toxicity into the aforementioned categories is taking place by calculating the following averages

$$E(\text{pt}) = \frac{\text{EP}(\text{etwc}) + \text{EP}(\text{etsc}) + \text{EP}(\text{htw}) + \text{EP}(\text{hts})}{n}$$

(persistent toxicity),

$$E(\text{et}) = \frac{\text{EP}(\text{etwa}) + \text{EP}(\text{etp})}{n} \quad (\text{ecotoxicity}),$$

$$E(\text{ht}) = \text{EP}(\text{hta}) \quad (\text{human toxicity})$$

where “ n ” is the number of impact potentials entering into the numerators, for which values have been calculated.

The normalized references for the above impact categories are given (Daniel and Pappis, 2003a,b) as well as the weighting factors according to the reference year and region (1990, Greece). Thus, the weighted impact potentials for the above categories are calculated

$$\begin{aligned} \text{WP}_{\text{HT}} &= \text{NF}_{\text{HT}} * \text{WEP}_{\text{HT}} = 1.28 \times 10^{-5} * 1.4 \\ &= 1.80 \times 10^{-5} \text{ equiv./person} \end{aligned} \quad (14)$$

$$\begin{aligned} \text{WP}_{\text{ET}} &= \text{NF}_{\text{ET}} * \text{WEP}_{\text{ET}} = 2.67 \times 10^{-4} * 1.11 \\ &= 2.96 \times 10^{-4} \text{ equiv./person} \end{aligned} \quad (15)$$

$$\begin{aligned} \text{WP}_{\text{PT}} &= \text{NF}_{\text{PT}} * \text{WEP}_{\text{PT}} = 5.54 \times 10^{-5} * 1.18 \\ &= 6.54 \times 10^{-5} \text{ equiv./person} \end{aligned} \quad (16)$$

$$\begin{aligned} \text{WP}_{\text{HT}} &= \text{NF}_{\text{HT}} * \text{WEP}_{\text{HT}} = 4.10 \times 10^{-8} * 1.4 \\ &= 5.74 \times 10^{-8} \text{ equiv./person} \end{aligned} \quad (17)$$

$$\begin{aligned} \text{WP}_{\text{ET}} &= \text{NF}_{\text{ET}} * \text{WEP}_{\text{ET}} = 4.40 \times 10^{-5} * 1.11 \\ &= 4.89 \times 10^{-5} \text{ equiv./person} \end{aligned} \quad (18)$$

$$\begin{aligned} \text{WP}_{\text{PT}} &= \text{NF}_{\text{PT}} * \text{WEP}_{\text{PT}} = 8.17 \times 10^{-3} * 1.18 \\ &= 9.64 \times 10^{-3} \text{ equiv./person} \end{aligned} \quad (19)$$

2.7. Solid wastes (land use)

A last category of environmental impacts regarding ecological impacts concerns solid wastes (land use). This includes a number of emissions of waste to be dumped in various ways. Waste for dumping is classified into four categories according to the type of landfilling or dumping, namely, bulk waste, hazardous waste, slag and ashes and radioactive waste. In Table 3 the normalized and weighted solid wastes for the batteries

Table 3
Solid wastes (1997) during batteries reverse supply (RC) and disposal (D) chain

Substance	Normalization reference unit for 1994, $E_{SW(j)}$ (kg/person/year)	Actual solid wastes production, $E_{SW(j)}$ (kg/person/year)		Normalized solid wastes, $NSW(j)$ person ⁻¹ $NSW(j) = \frac{E_{SW(j)}}{E_{SWGR(j)}}$		Weighted factor for solid wastes WEP(j)	Weighted solid waste j , WSW(j) WSW(j) = WEP(j) * NSW(j)	
		RC	D	RC	D		RC	D
Bulk waste	380	0.313	0.313	8.25×10^{-4}	8.25×10^{-4}	1.1	9.07×10^{-4}	9.07×10^{-4}
Slag and ashes	235	1.03	–	4.38×10^{-3}	–	1.1	4.82×10^{-3}	0.0
Hazardous waste	30	0.023	1.27	7.69×10^{-4}	4.23×10^{-2}	1.1	8.46×10^{-4}	4.65×10^{-2}
Radioactive waste	0.035	0.0	0.0	0.0	0.0	1.1	0.0	0.0

Table 4
Resources consumption (1997) during the batteries reverse supply chain (World Resources Institute, 1990)

Substance	Normalization reference unit for 1990, $E_{RCW(j)}$ (kg/person/year)	Actual resources consumption, $E_{RC(j)}$ (kg/person/year)	Normalized resource consumption NR(j) person ⁻¹ $NR(j) = \frac{E_{RC(j)}}{E_{RCW(j)}}$	Weighted factor for resource consumption, WEP(j)	Weighted consumption for resource j , WR(j) $WR(j) = WEP(j) * NR(j)$
Carbon	570	1.94×10^{-4}	3.4×10^{-7}	0.0058	1.66×10^{-9}
Oil	590	3.12×10^{-4}	5.29×10^{-7}	0.023	1.22×10^{-8}
Iron	100	2.33×10^{-4}	23.3×10^{-7}	0.0085	19.8×10^{-9}

Table 5
Resources consumption (1997) during the batteries disposal chain (World Resources Institute, 1990)

Substance	Normalization reference unit for 1990, $E_{RCW(j)}$ (kg/person/year)	Actual resources consumption, $E_{RC(j)}$ (kg/person/year)	Normalized resources consumption, NR(j) person ⁻¹ $NR(j) = \frac{E_{RC(j)}}{E_{RCW(j)}}$	Weighted factor for resources consumption, WEP(j)	Weighted resources consumption j , WR(j) $WR(j) = WEP(j) * NR(j)$
Oil	590	2.21×10^{-4}	3.74×10^{-7}	0.023	8.61×10^{-9}
Lead	0.64	1.36×10^{-3}	2.13×10^{-3}	0.048	1.02×10^{-4}
Copper	1.7	1.62×10^{-6}	9.53×10^{-7}	0.028	26.6×10^{-9}
Antimony		9.73×10^{-6}	9.73×10^{-6}	1.0	9.73×10^{-6}

reverse supply and disposal chains are estimated. Due to the fact that the environmental impacts of the solid wastes have local consequences, the region of Greece is used as normalization reference.

3. Resources consumption

“Resources consumption” represents one of the three major environmental impact categories examined in this study. Impact assessment evaluates resources consumption inventoried in the examined EOL scenarios. Thus, the concentration of the inventoried data concerning consumption of raw materials, ancillary materials and fuel as well as resources are expressed as pure substances and not as ore. This task has been in part performed in the stage of Inventory Analysis of the batteries reverse supply and disposal chain.

Comparing the alternative EOL scenarios of used batteries, it is clear that the main resources consumption takes place in the batteries reverse supply chain. In Table 4 the final weighting of the inventoried resources consumed in the batteries reverse supply chain appears.

However, in the case of the batteries disposal chain we assume that the quantities of lead, copper and antimony, which could be recovered during the reverse supply chain, are “consumed”. So,

in Table 5 the weighting resources “consumed” in the batteries disposal chain are presented (except of oil, which is actually consumed as fuel in transportation of used batteries to landfills).

4. Impacts on the working environment

The last of the major environmental impact categories examined in this study concerns “impacts on the working environment”. In this category many impacts depend on the type of the environmental exchanges and the way of exposure.

In this study no data was available for both the batteries reverse supply and the disposal chains. Therefore, the evaluation of their impacts was not feasible.

5. Discussion and conclusions

In most cases an LCA comparison between materials, products or processes aims at identification of the best ones (Hanssen et al., 1994). This is intended for reasons of marketing and environmental policies. In this paper, a comparison of the alternative EOL scenarios of logistics channels of disposal and recovery of used batteries has been performed. This comparison refers to the environmental impacts produced from these channels and its goal is to investigate the necessity of the creation and fur-

ther development of an EOL scenario such as the reverse supply chain of batteries.

In the inventory table presented in the previous phase of LCA, it is shown which substances are exchanged with the surroundings as a result of the processes, which take place in the batteries reverse supply and disposal chains (Daniel and Pappis, 2003a,b). An overview of where the most important exchanges (in terms of quantities of substances) occur is also provided. The exchanges were summarized in resources consumption, power demands and emissions to air, water and soil as wastes. Impacts on the working environment are not accounted in this case study due to lack of data. The exchanges with the environment, which result from the batteries reverse supply and disposal chains, are converted to environmental impact potentials according to the methods described earlier. It must be noted that not all the exchanges result in impacts on the environment while the large quantities of substances that are released do not necessarily have a respective contribution to environmental impacts. This is explained by the fact that different substances, among others, possess different hazard properties. So, an impact assessment of the individual exchanges is required in order to identify the most significant environmental impacts in each EOL scenario.

The final results of this impact assessment are presented in Appendix A (Figs. A.1 and A.2). The weighted impact potentials expressed in person equivalents for each environmental category are shown graphically. The results of the impact assessment indicate several points for discussion according to the major environmental impact categories. Examining each one of them, the following conclusions can be drawn.

5.1. Global warming

The reverse supply chain of batteries demands a considerable quantity of energy in the form either of electrical energy or liquid fuels consumed. This quantity is consumed mainly within lead-processing facilities (life cycle stages of disaggregation and

recycling of used batteries). However, the contribution to global warming of the greenhouse gases released as result of the energy consumed is very low in relation to greenhouse gases that are released during the collection–distribution processes (Fig. 1). This can be explained by the extended use of transportation means (particularly sea transportation, which is responsible for large emissions of greenhouse gases).

Not unexpectedly, the batteries reverse supply chain consumes much more energy as compared to the batteries disposal chain. This consumption has direct impact to the increase in global warming as shown in Fig. 1. The batteries reverse supply and disposal chains contribution can be also seen in relation to a person's average contribution. So, the contribution of the reverse supply chain is equal to 17μ equiv./person and of the disposal chain to 0.00865μ equiv./person. These normalized values correspond to $1.7 \mu\%$ and $0.000865 \mu\%$ of the contributions for an average person in 1994, respectively. Clearly, the values corresponding to the EOL scenarios examined are quite different (1960 times). This difference can be also shown in the Impact Table (Table 1) of the weighted impact potentials of the examined EOL scenarios.

5.2. Stratospheric ozone depletion

The issue of impacts on ozone depletion for both reverse supply and disposal chains of batteries has not been addressed. This is attributable to the fact that there are no inventoried substances, which contribute to stratospheric ozone depletion, as said earlier.

5.3. Photochemical ozone formation

The most significant impacts inventoried in this category are coming from exchanges of carbon monoxide (CO) and volatile organic compounds (VOCs) from vehicles, which are used mostly in the collection–distribution processes either in the batteries reverse supply or the disposal chains.

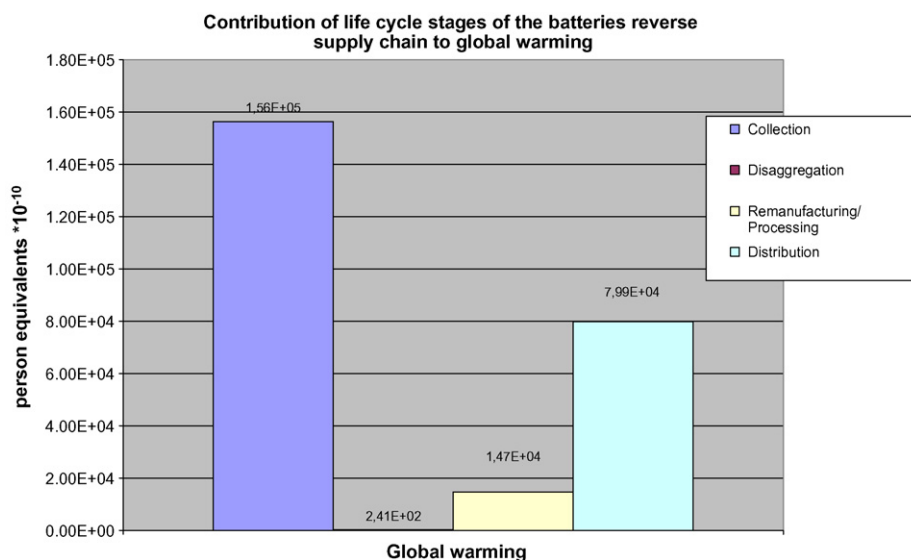


Fig. 1. Contribution of life cycle stages of the batteries reverse supply chain to global warming.

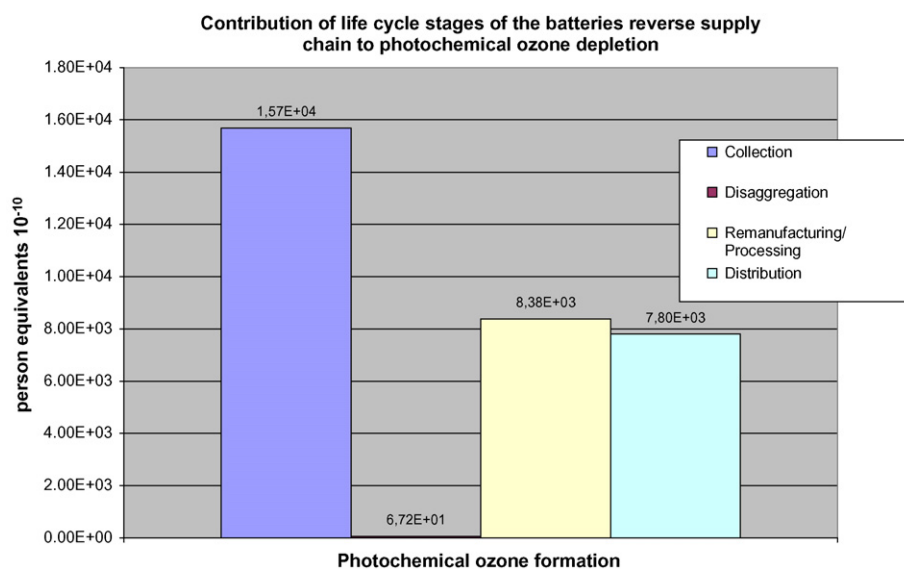


Fig. 2. Contribution of life cycle stages of the batteries reverse supply chain to photochemical ozone depletion.

As shown in Fig. 2, the contribution of transportation means (vehicles) to photochemical ozone formation is evident. Their extended use in the collection–distribution processes of the batteries reverse supply chain contributes heavily in this environmental category. Instead of this, the contribution of lead-processing facilities (life cycle stages of disaggregation and recycling) is less significant ($\approx 26.5\%$).

From the comparison of the examined EOL scenarios (Fig. 1) the small contribution of the batteries disposal chain as compared to the reverse supply chain (0.87 times) to photochemical ozone formation is evident. This is explained by the use of sea transportation means in the reverse supply chain of batteries, which are responsible for some additional emissions of photo-oxidant gases such as formaldehyde (HCHO) that are not emitted in the collection–distribution processes of the batteries disposal chain.

5.4. Acidification

As shown in Fig. 3, the great contribution ($\approx 64.2\%$) to acidification occurs from substances released mainly in the lead-processing facilities. This is attributable to the fact that a large quantity of sulfur dioxide (SO_2) and nitrogen oxides (NO_x) is released during the recovery process of lead in the rotary furnace and less during the separation of batteries in their components (disaggregation stage), as it was expected. This happens because of large quantities of acid contained in them. Acid contributes heavily to acidification. However, the process of acid neutralizing, as well as the fact that a significant quantity of (imported) batteries are empty of acids, takes place only in the batteries reverse supply chain. This means that smaller quantities of acidifying substances are released during the batteries reverse supply than the disposal chains. As a result of this, the weighted impact

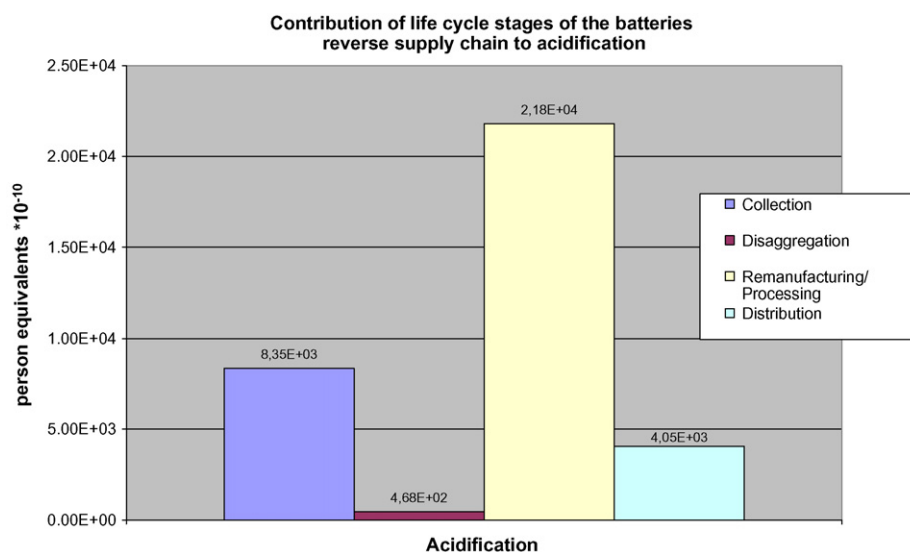


Fig. 3. Contribution of life cycle stages of the batteries reverse supply chain to acidification.

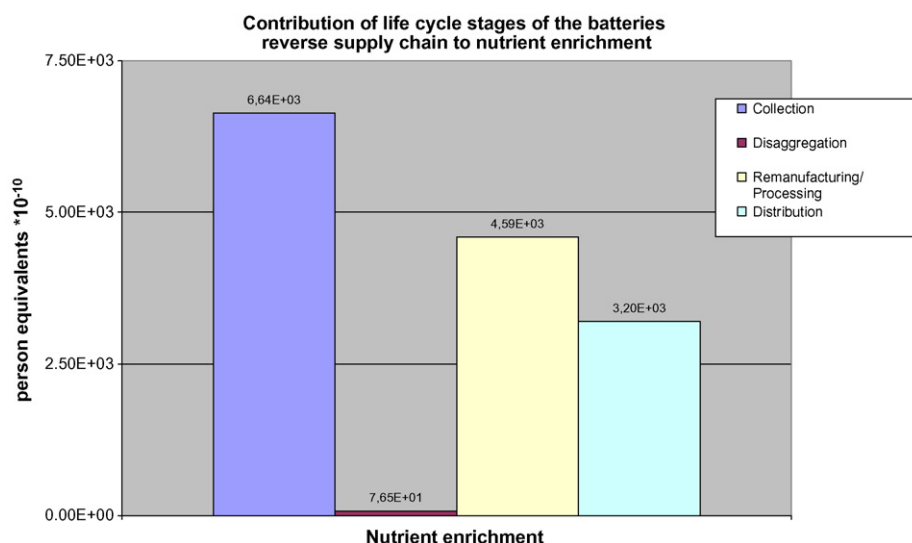


Fig. 4. Contribution of life cycle stages of the batteries reverse supply chain to nutrient enrichment.

potentials of the disposal chain accounts for at least two orders of magnitude (38.6 times) larger than those of the reverse supply chain of batteries (Fig. 1).

5.5. Nutrient enrichment

The release of nitrogen oxides along the batteries reverse supply chain stages is a basic reason for nutrient enrichment. These emissions are coming mainly from the transportation and lead recovery processes. The most important contribution is observed in the life cycle stage of batteries collection. This is explained by the extended use of trucks (>16 t) responsible for large quantity emissions of nitrogen oxides (Fig. 4).

Additionally, the larger use of transportation means responsible for increased nitrogen oxides emissions (trucks > 16 t) by the batteries disposal chain compared to the reverse supply chain result in larger environmental implications in this category. More particularly, the weighted impact potentials of the disposal chain are estimated about 1.15 times larger than that of the reverse supply chain (Fig. 1).

5.6. Ecotoxicity–human toxicity

Examining the main characterization factors of a toxic substance either for ecosystems or human health, the high toxic effects of the significant quantities of acid and lead contained in batteries are evident. By the comparison of these substances, the toxic effects of lead are classified as more dangerous than the acid ones. However, the ecotoxicity impact potentials of the disposal chain are six times smaller than the reverse supply chain ones, as is shown in Fig. 1. This is attributable to the fact that, in spite of the large quantity of lead and acids that are released in the disposal chain, an additional substance (formaldehyde) that is released in the reverse supply chain as a result of the transportation means is contributing heavily to ecotoxicity.

A great contribution in the reduction of persistent toxicity impact potentials is due to the reverse supply chain as a result

of lead recovery, as compared to the disposal in landfills. The persistent toxicity impact potentials could be smaller if there were a more efficient recovery system, which would reduce the lead discharges.

The toxic properties of lead have been known for thousands of years and this has led to strict environmental regulations. Lead in blood levels is higher for the employees at lead handling plants and the people who live near in these plants than among the average citizen (Socolow and Thomas, 1997). Unexpectedly, the difference between human toxicity potential impacts for the reverse supply and disposal chains was enormous (at least four orders of magnitude). This can be explained by the considerable quantity of lead, which is emitted through air in the lead-processing facilities.

5.7. Solid wastes (land use impacts)

In this ecological category, the main difference between the examined EOL scenarios concerns the quantity of lead in the form of solid wastes that is disposed of in landfills associated with the well-known hazards against human health and nature. In the reverse supply chain, a significant quantity of lead can be recovered. The magnitude of this quantity may change according to the recovery technology applied. Regarding the slag and ashes category, the solid wastes concern the remains of lead-processing facilities, while the components of batteries, such as separators, grips, and cases made of plastics, are classified as bulk wastes forwarded to landfills for both EOL scenarios.

As shown in Table 1, the relative importance of these solid waste categories in relation to the other impact categories is evident. This is attributable to the fact that a lot of quantities of solid wastes are produced and handed out for both EOL scenarios. The contribution of the batteries reverse supply chain for hazardous waste corresponds to 0.00077% of the average contribution of a person in the reference year of 1994. The respective percentage for the batteries disposal chain is estimated to 0.0423%. This was predictable as long as a large quantity of the hazardous waste

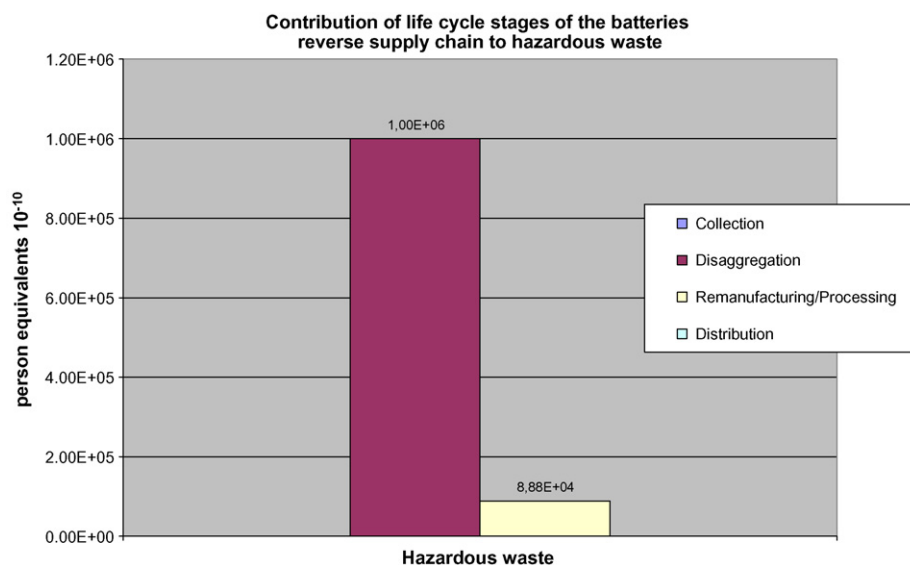


Fig. 5. Contribution of life cycle stages of the batteries reverse supply chain to hazardous waste.

(i.e. lead) is going for recycling in the case of reverse supply chain. Regarding the batteries reverse supply chain (Fig. 5), it is evident that the main source of hazardous wastes is the stage of disaggregation. This happens because of lead discharges during the drying process of lead paste.

5.8. Resources consumption

The resources consumption is weighted in relation to the size of known global reserves. The weighting process ranks the resources consumption on the basis of a scarce resources criterion (i.e. resources which are most threatened by depletion). Resources consumption takes place for both EOL scenarios examined. In the reverse supply chain, raw and secondary raw materials are used for the recycling of lead (Ayres, 1995; Wilson, 1993) as well as consumed fuels (Fig. 2). In the disposal chain, the quantity of lead, which is not recovered, is assumed to contribute to its depletion.

It is noticed that lead is ranked as the most scarce resource for both scenarios. However, in the EOL scenario of reverse supply chain less quantity of lead is “consumed” as a result of the lead recovery process (Fig. 2). This “consumed” quantity of lead could be smaller if there were a possibility to increase the low efficiency of the current recycling unit ($\approx 40\%$) and to improve the quality of the used batteries delivered.

It is also noted that the quantities of copper and antimony contained in batteries may be recovered during the reverse supply chain. As shown in Fig. 2, these quantities are also assumed to be “consumed” in the batteries disposal chain. Finally, it can be concluded that lead is of the highest priority in relation to the rest of the resources.

5.9. Impacts on the working environment

Due to lack of data, no impacts on the working environment are addressed in this study.

In conclusion, it can be argued that recovery of lead from used batteries has many advantages not only from an economic perspective (Zabaniotiu et al., 1999) but also from an environmental one. The overall environmental profile of such an endeavor, i.e., the operation of batteries reverse supply chain is shown to be more satisfactory compared to the disposal of batteries. Analyzing the results of the impact assessment phase, a relative equivalence between the examined scenarios is evident. However, this may not be enough for drawing a final conclusion about which is the best solution. So, the need for implementation of appropriate ranking and evaluation methods, which may produce safe and solid results concerning the choice about the best solution, is evident (Domingos et al., 2004). Such methods could be different valuation methods (e.g. multicriteria analysis, AHP, aggregation methods (Daniel et al., 2004; Domingos et al., 2004), etc.).

Additionally, the proposed LCIA method is strongly recommended in case an environmental engineer is trying to compare the behavior of the examined EOL scenarios under different weighting factors. It is well known that the importance of the impact categories is changing dynamically during time. The local, regional and global environmental priorities are changing since new scientific findings are coming into the surface. Impact categories, which have been ranked highly environmentally dangerous in the previous years, are now classified as less important whether the society made the necessary actions to reduce their consequences or some other impact categories have been classified as more important than them. A remarkable example is the importance of the impact category of stratospheric ozone depletion compared to global warming. During the last years, a worldwide applied policy, aiming to reduce significantly the volume of the substances contributing to stratospheric ozone depletion, has been adopted. This has lead to a drastic reduction of the impacts and to change the relative importance against the other impact categories. At the same time, the importance of global warming is becoming even greater since its grave impacts

are not only present but also are increasing steadily even though an emissions reduction policy has been applied worldwide.

Notwithstanding the change of the weighting factors, an environmental engineer can still very easily extract new conclusions about each EOL scenario by applying the new weighting factors in the analysis previously applied. Thus, the environmental profile of the examined EOL scenarios can be changed and the overall conclusions for each channel can be altered accordingly.

The proposed analytical approach can be also used as a database for future comparisons between the same impact categories of each EOL scenarios of used lead-acid batteries. This is also useful to environmental engineers in case of comparisons

among the best available technologies used in the same EOL scenarios.

Acknowledgements

The research presented in this paper has been supported by the Greek ministry of Education and the European Union.

Appendix A. Appendix A

See Figs. A.1 and A.2.

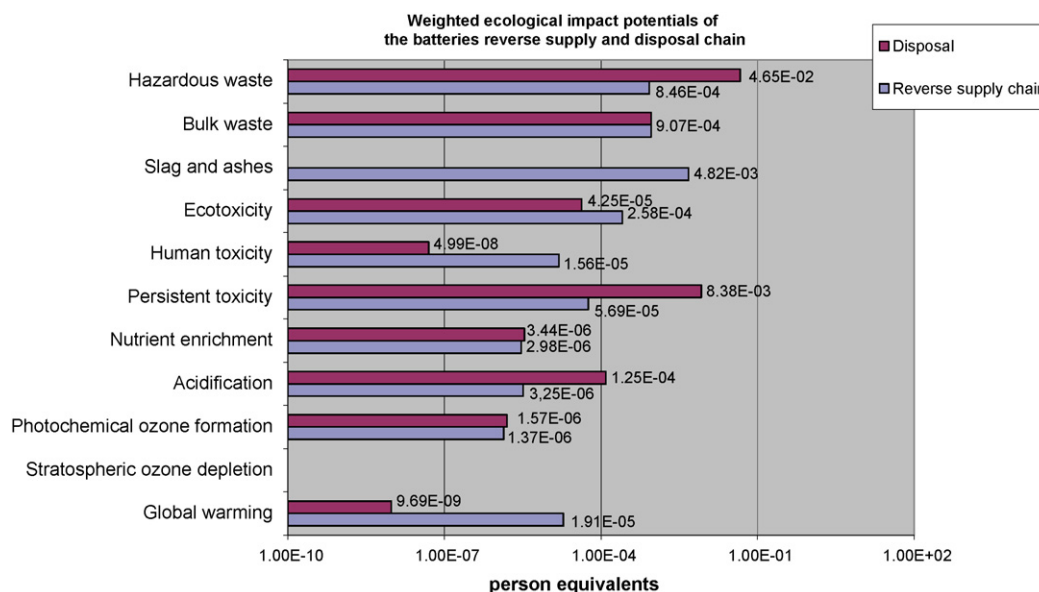


Fig. A.1. Weighted ecological impact potentials of the batteries reverse supply and disposal chain.

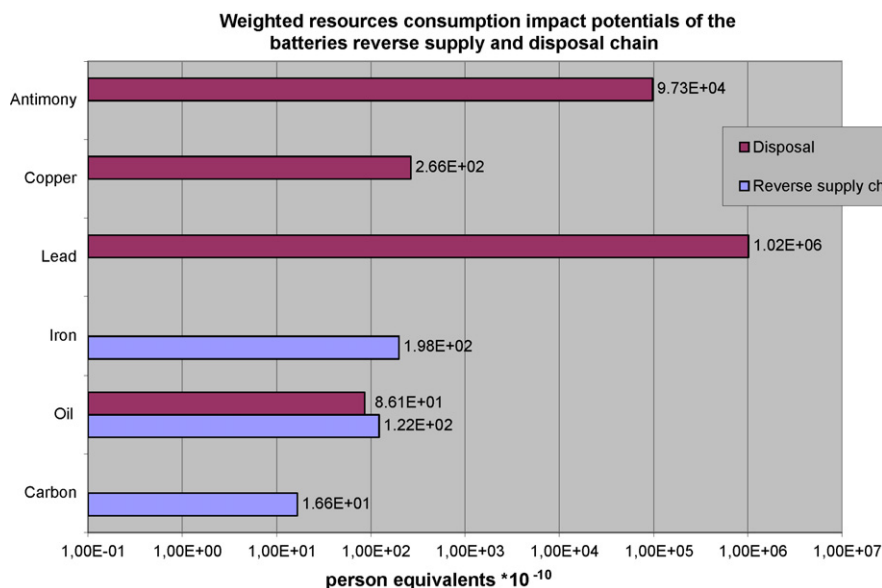


Fig. A.2. Weighted resources consumption impact potentials of the batteries reverse supply and disposal chain.

References

- Abhe S, Braunschweig A, Muller-Wenk R. Methodik für Ökobilanzen auf der Basis ökologischer Optimierung. Schriftenreihe Umwelt Nr. 133. Bern: Bundesamt für Umwelt, Wald und Landschaft (BUWAL); 1990.
- Andersson-Skold Y, Grennfelt P, Pleijel K. Photochemical ozone creation potentials: a study of different concepts. *J Air Waste Manage Assoc* 1992;42(9):1152–8.
- Ayres RU. Life cycle analysis: a critique. *Resour, Conserv Recycl* 1995;14:199–223.
- Berkhout F. Life cycle assessment and industrial innovation. In: The 1995 eco-management and auditing conference; 1995. p. 50–64.
- Bringezu S, Hinterberger F, Schutz H. Integrating sustainability into the system of national accounts: the case of interregional material flows. In: Int. Afcet Symp. Models of sustainable development—exclusive or complementary approaches to sustainability; 1994. p. 669–80.
- Busch NJ. Calculation of weighting factors. In: Stranddorf HK, Hoffmann L, Schmidt A, editors. Update on impact categories, normalization and weighting in LCA. Environmental Project no. 995, Danish EPA; 2005.
- Curran MA. Broad-based environmental life cycle assessment. *Environ Sci Technol* 1993;27(3).
- Daniel SE, Pappis CP. Applying life cycle inventory analysis to reverse supply chains: a case study. *Resour, Conserv Recycl* 2003a;37(4):251–340.
- Daniel SE, Pappis CP. Applying life cycle inventory analysis to reverse supply chains: a case study. Working Paper, University of Piraeus; 2003.
- Daniel SE, Tsoulfas TG, Rahaniotis N, Pappis CP. Aggregating and evaluating the results of different environmental impact assessment methods. *Ecol Indic* 2004;4(2):125–38.
- Domingos T, Domingos JJD, Simões A, Sousa T, Vieira R. Task 3-aggregation methods. EXTENSITY: environmental and sustainability management systems in extensive agriculture, LIFE03 ENV/P/505; 2004.
- Goedkoop M. The Eco-indicator 95. Final Report, NOH report 9523, Pre Consultants, Amersfoort, The Netherlands; 1995.
- Groupe des Sages. Guidelines for the application of life cycle assessment in the EU eco-label award scheme. A report prepared for the European Commission by the Groupe des Sages. Office for Official Publications of the European Communities; 1997. ISBN 92-827-8684-6.
- Hanssen OJ, Forde JS, Thoresen J. Environmental indicator and index systems. An overview and test of different approaches. A pilot study for Statoil. Ostfold Research Foundation 1994, Research Report OR 17.94;1994.
- Hanssen OJ. Environmental impacts of product systems in a life cycle perspective: a survey of five product types based on life cycle assessment studies. *J Clean Prod* 1998;6:299–311.
- Hauschild M, Wenzel H. Environmental assessment of products, scientific background, vol. 2. Chapman and Hall; 1998.
- Heijungs R, Guinee JB, Huppes G, Lankreijer RM, Udo des Haes HA, Wegener Sleeswijk A, et al. Environmental life cycle assessment of products. Guide and backgrounds. Leiden, The Netherlands: CML, Leiden University; 1992.
- Hertwich EG, James KH. A Decision-analytic framework for impact assessment. Part 2. Midpoints, endpoints, and criteria for method development. *Int J LCA* 2001;6.
- Hertwich EG, Pease WS, Koshland CP. Evaluating the environmental impact of products and production processes: a comparison of six methods. *Sci Total Environ* 1997;196:13–29.
- Hinterberger F, Kranendonk S, Welfens MJ, Schimdt-Bleek F. Increasing Resource productivity through Eco-efficient Services. Wuppertal Papers Nr. 13. Wuppertal institute, Duppertsberg 19, 42103 Wuppertal, Germany; 1994.
- Houghton JT, Jenkins GJ, Ephraums JJ, editors. Climate change. The IPCC scientific assessment. Cambridge University Press; 1990.
- Houghton JT, Callander BA, Varney SK, editors. Climate change 1992. The supplementary report to the IPCC scientific assessment. Cambridge University Press; 1992.
- ISO 14040:1997 (E). Environmental management-life cycle assessment-Principles and Framework; 1997.
- ISO 14041:1998 (E). Environmental management-life cycle assessment-goal and scope definition and inventory analysis; 1998.
- ISO 14042:2000 (E). Environmental management-life cycle assessment-life cycle impact assessment; 2000.
- Krozer J, Vis JC. How to get LCA in the right direction. *J Clean Prod* 1998;6:53–61.
- Lee JJ, O'Callaghan P, Allen D. Critical review of life cycle analysis and assessment techniques and their application to commercial activities. *Resour, Conserv Recycl* 1995;13:37–56.
- Lindfors Lars GK, Christiansen L, Hoffman L, Virtanen Y, Juntilla V, Hanssen OJ, et al. LCA-NORDIC, Technical Reports No 10 and Special Reports No 1–2; 1995b.
- Lindfors Lars-GK, Christiansen L, Hoffman L, Virtanen Y, Juntilla V, Hanssen OJ, et al. LCA-NORDIC, Technical Reports No 1–9; 1995a.
- Miettinen P, Hamalainen RP. How to benefit from decision analysis in environmental life cycle assessment (LCA). *Eur J Operat Res* 1997;102:264–75.
- Narodoslawsky RB, Krotscheck C. The sustainable process index (SPI): evaluating processes according to environmental compatibility. *J Hazard Mater* 1995;41(2/3):383–97.
- NORD (Nordic Council of Ministers). Nordic guidelines on product life-cycle assessments. 1995; 20.
- Rydh CJ. Environmental assessment of vanadium redox and lead-acid batteries for stationary energy storage. *J Power Sources* 1999;80:21–9.
- Sage J. Industrielle Abfallvermeidung und deren Bewertung am Beispiel der Leiterplattenherstellung. Austria: dbv-Verlag, technique Universität Graz; 1993.
- Schmidt-Bleek F. Wieviel Umwelt braucht der Mensch? Mips-Das Maß für ökologisches Wirtschaften. Berlin: Birkhauser Verlag GmbH; 1994.
- SETAC (Society of Environmental Toxicology and Chemistry) and SETAC Foundation for Environmental Education. Inc. A technical Framework for life-cycle assessments; January 1991, p. xviii.
- SETAC (Society of Environmental Toxicology and Chemistry) and SETAC Foundation for Environmental Education Inc. Guidelines for life-cycle assessment: code of practice. Brussels: Asetac; 1993.
- Socolow R, Thomas V. The industrial ecology of lead and electric vehicles. *J Indust Ecol* 1997;1(1):13–36.
- Solomon S, Albritton DL. Time-dependent ozone depletion potentials for short- and long-term forecasts. *Nature* 1992;357:33–7.
- Srinivasan M, Wu T, Sheng P. Development of a scoring index for the evaluation of environmental factors in machining processes. Part 1. Health hazard score formulation. *Trans NAMRI/SME* 1992;23:115–22.
- Stranddorf HK, Hoffmann L, Schmidt A. Impact Categories, Normalization and weighting in LCA. Danish Ministry of Environment, EPA. Environmental News, No 78; 2005.
- Tellus Institute. The Tellus Packaging Study. Boston, MA, USA: Tellus Institute; 1992.
- Tørsløv J. Ecotoxicity. In: Stranddorf HK, Hoffmann L, Schmidt A, editors. Update on impact categories, normalization and weighting in LCA. Environmental Project no. 995, Danish EPA, 2005.
- UNECE. Revised draft technical annex on classification of volatile organic compounds issued on their photochemical ozone creation potential (POCP). United Nations Economic and Social Council, Economic Commission for Europe. EB.AIR/WG.4/R.13/Rev; 1991a.
- UNECE. Protocol to the 1979 convention on long-range transboundary air pollution concerning the control of emissions of volatile organic compounds or their transboundary fluxes. United Nations Economic Commission for Europe. ECE/EB.AIR/30; 1991b.
- UNECE. Major review on strategies and policies for air pollution abatement, tables and figures. EB.AIR/R.87/Add. 1. 21 September 1994. Geneva: United Nations Economic Commission for Europe; 1994.
- Wenzel H, Hauschild M, Alting L. Environmental Assessment of Products. Methodology, vol. 1. Chapman and Hall; 1997.
- Wilson DN. Recycling of lead, advances in recovery and technology, concepts and technology. In: Collected paper for the ReC'93. International Recycling Congress; 1993. p. 258–66.
- World Resources Institute. World Resources 1990-1991: a guide to the global environment. A report by the World Resources Institute in collaboration with the United Nations Environment Programme and the United Nations Development Programme. Oxford University Press; 1990.
- Zabanioti A, Kouskoumelaki E, Sanopoulos D. Recycling of spent/acid batteries: the case of Greece. *Resour Conserv Recycl* 1999;25:301–17.